

Source-Related Transport of Phosphorus in Surface Runoff

Francirose Shigaki, Andrew Sharpley,* and Luís Ignácio Prochnow

ABSTRACT

Continual application of mineral fertilizer and manures to meet crop production goals has resulted in the buildup of soil P concentrations in many areas. A rainfall simulation study was conducted to evaluate the effect of the application of P sources differing in water-soluble P (WSP) concentration on P transport in runoff from two grassed and one no-till soil (2 m² plots). Triple superphosphate (TSP)–79% WSP, low-grade single superphosphate (LGSSP)–50% WSP, North Carolina rock phosphate (NCRP)–0.5% WSP, and swine manure (SM)–30% WSP, were broadcast (100 kg total P ha^{−1}) and simulated rainfall (50 mm h^{−1} for 30 min of runoff) applied 1, 7, 21, and 42 d after P source application. In the first rainfall event one d after fertilizer application, dissolved reactive P (DRP) and total P (TP) concentrations of runoff increased ($P < 0.05$) for all soils with an increase of source WSP; with DRP averaging 0.27, 0.50, 14.66, 41.69, and 90.47 mg L^{−1}; and total P averaging 0.34, 0.61, 19.05, 43.10, and 98.06 mg L^{−1} for the control, NCRP, SM, LGSSP, and TSP, respectively. The loss of P in runoff decreased with time for TSP and SM, such that after 42 d, losses from TSP, SM, and LGSSP did not differ. These results support that P water solubility in P sources may be considered as an indicator of P loss potential.

FRESHWATER EUTROPHICATION IS A MAJOR WATER QUALITY CONCERN in the USA (USEPA, 1996) and worldwide (National Research Council, 2000), and can be greatly accelerated by the influx of phosphorus (P) in surface runoff from agricultural land. Eutrophication restricts water use for fisheries, recreation, and industry due to the increased growth of undesirable algae and aquatic weeds, and oxygen shortages caused by their death and decomposition. Also, an increasing number of surface waters have experienced periodic and massive harmful algal blooms (e.g., *cyanobacteria* and *Pfiesteria*), which contribute to summer fish kills, unpalatability of drinking water, formation of carcinogens during water chlorination, and links to neurological impairment in humans (Burkholder and Glasgow, 1997).

As P most often limits freshwater eutrophication, decreasing P loss in runoff has become a major target in minimizing surface water degradation (Carpenter et al., 1998; U.S. Geological Survey, 1999). To meet crop production goals, however, P is routinely added to soil. The continued application of fertilizers and manures in many areas has resulted in the buildup of soil P concentrations

above those required for optimum plant growth. These elevated concentrations increase the potential for P loss (Sims et al., 1998; Sharpley, 2003).

There are many types of P fertilizers available, such as organic (bone meal, cottonseed meal, fish emulsions, manures, and processed sewage sludge) and inorganic (reactive rock phosphates, acidulated phosphates, and thermophosphates) materials. The most common P fertilizer used is acidulated triple superphosphate (TSP) due its high P solubility and thus, immediate availability of P for crop uptake. Other fertilizers with lower water P solubility, such as highly reactive rock phosphate (RP) and low-grade single superphosphate (LGSSP), can be used to maintain a long-term source of P due to their residual availability. Among RP, NCRP (North Carolina RP) and GRP (Gafsa RP) are widely applied to agricultural soils due to their high reactive content of P and high agronomic effectiveness (Leon et al., 1986; Chien and Menon, 1995). Low-grade single superphosphate has been tested at greenhouse and field condition with good performance in many agronomical scenarios (Prochnow et al., 2003; Prochnow et al., 2004). This P source is produced from low-grade apatite concentrate with the objective of optimizing the use of RP.

Recent studies have shown a close relationship between the water solubility of P in land-applied manures and runoff P. Kleinman et al. (2002) found the WSP concentration of dairy, poultry, and swine manure applied to the surface of three soils to be highly correlated with dissolved reactive P (DRP) in surface runoff ($r^2 = 0.86$, $P < 0.01$). DeLaune et al. (2004) and Withers et al. (2001) observed DRP losses from P-amended soils were proportional to the water-soluble content of applied P. However, few studies compared the effect of P fertilizers varying in WSP on the transport of P in agriculture runoff.

Sharpley et al. (1978) evaluated the effects of applying dicalcium phosphate (DCP–4% WSP) and single superphosphate (SSP–75% WSP) at a rate of 50 kg P ha^{−1} on the transport of DRP and particulate P (PP) in runoff from plots (55 m²) under pasture and natural rainfall 15 wk after application. They observed a slightly greater proportion of applied SSP was transported as DRP (5%) than with DCP (3%). This difference was attributed to more rapid dissolution of SSP than of DCP at the soil surface. The greater proportion of applied DCP lost as PP (8%) than with SSP (4%) was attributed to an increased loss of P by washoff of less soluble DCP particles (Sharpley et al., 1978). Consequently, applica-

F. Shigaki and L.I. Prochnow, Dep. Ciência do Solo, ESALQ, Univ. of São Paulo, Piracicaba, SP, Brazil. A. Sharpley, USDA-ARS, Pasture Systems and Watershed Management Research Unit, Curtin Rd., University Park, PA 16802 USA. Mention of trade names does not imply endorsement by the U.S. Department of Agriculture. Received 20 Mar. 2006. *Corresponding author (Andrew.sharpley@ars.usda.gov).

Published in J. Environ. Qual. 35:2229–2235 (2006).

Technical Reports: Surface Water Quality
doi:10.2134/jeq2006.0112

© ASA, CSSA, SSSA

677 S. Segoe Rd., Madison, WI 53711 USA

Abbreviations: DCP, dicalcium phosphate; DRP, dissolved reactive P; LGSSP, low-grade single superphosphate; NCRP, North Carolina rock phosphate; PP, particulate P; RP, rock phosphate; SM, swine manure; SSP, single superphosphate; TP, total P; TSP, triple superphosphate; WSP, water-soluble P; GRP, Gafsa rock phosphate; RRP, reactive rock phosphate.

tion of DCP resulted in greater total P (TP) loss (7.09 kg ha^{-1}) in runoff than did an equivalent application of SSP (5.63 kg ha^{-1}). These results indicated that SSP is more susceptible to loss in runoff as DRP than from DCP, as a result of a more rapid and extensive dissolution. However, few studies have compared a range of fertilizer materials and manures varying in WSP on the transport of P in runoff. This information is needed to better define the agronomic effectiveness of P sources applied to agricultural land as well as environmental risk. Related to P loss risk assessment, the P Index that has recently been adopted by 49 states in the USA to address P-based nutrient management plans, considers water solubility of applied P as a factor influencing the potential for P runoff (Sharpley et al., 2003; USDA and USEPA, 1999). Phosphorus source solubility has focused on organic amendments (i.e., manure), with little consideration of the relative runoff P enrichment as a function of mineral fertilizer type added (Kleinman et al., 2002).

This paper describes a study of the effect of applying different P sources varying in P solubility on the trans-

port of P in runoff from three soils in southcentral Pennsylvania which were subject to simulated rainfall following the protocol of the National Phosphorus Research Project (2006). It is important to make the point that in this present study mineral P sources and swine manure (usually defined as an organic source) were used. However, we will refer to both mineral and organic P additions as “sources of P” to facilitate understanding of our discussion.

MATERIALS AND METHODS

Soils and Plot Location

This study was conducted in the mixed land-use watershed, FD-36 (Northumberland, south central Pennsylvania), a 39.5-ha subwatershed in the Mahantango Creek watershed, which drains into the Susquehanna River and ultimately the Chesapeake Bay (Fig. 1). Three sites in FD-36 were used; Alvira (Aeric Fragiaquilt) in orchardgrass (*Dactylis glomerata* L.), Berks (Typic Hapludults) under no-till corn (*Zea mays* L.), and Watson (Typic Fragiudults) in orchardgrass. The orchardgrass plots were hayed rather than grazed.

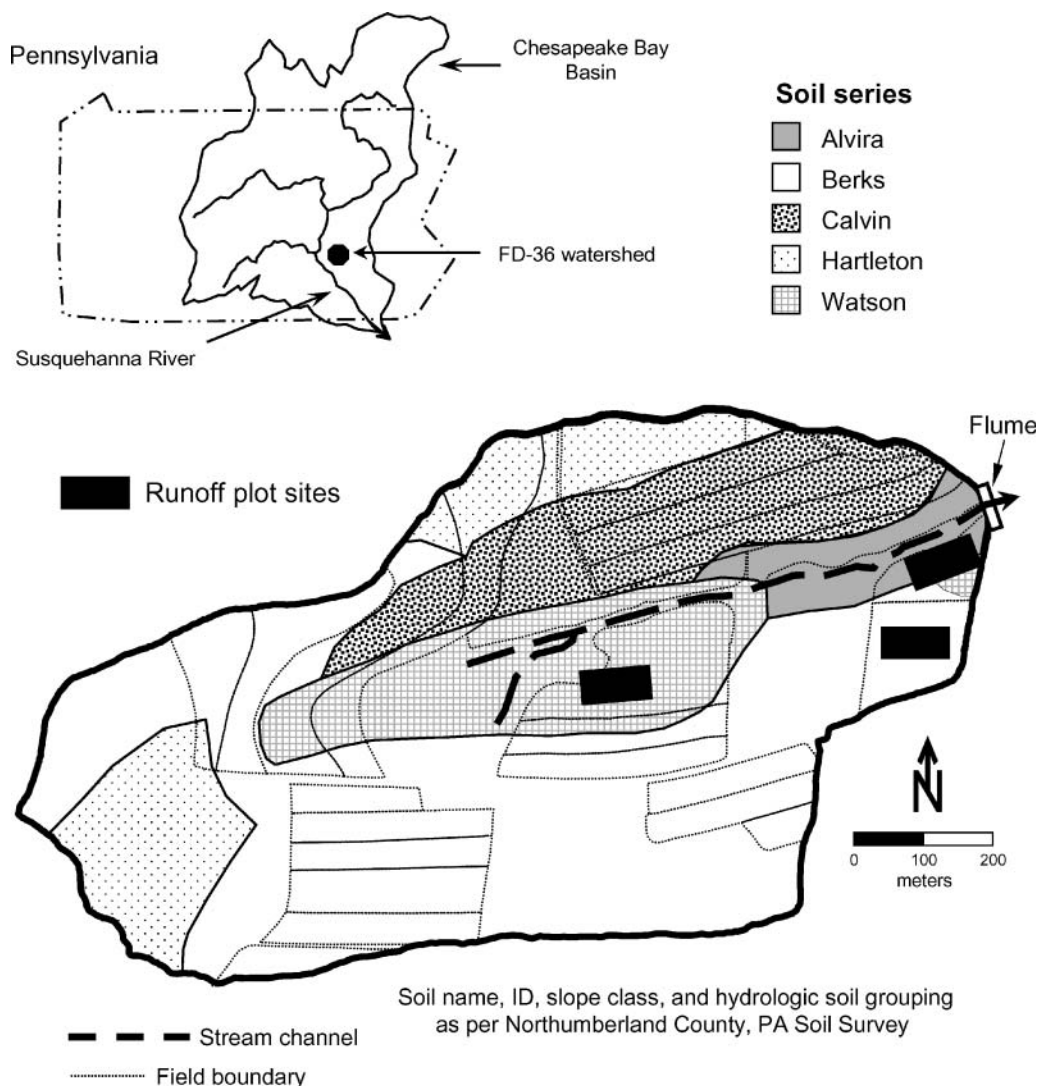


Fig. 1. Location, field boundaries, and soil types of the FD-36 watershed, Northumberland County, PA and field overland flow sites for this study.

Phosphorus Sources

Three mineral fertilizer P sources varying in their chemical properties with emphasis in the water solubility and swine manure were selected for this study. Fertilizers used were LGSSP (obtained by the acidulation of Araxa rock phosphate), TSP, and NCRP. Water solubility of mineral sources was determined by Association of Official Analytical Chemists Official Method 977.01; and ammonium citrate-EDTA solubility by Method 993.31 (Association of Official Analytical Chemists, 1995).

For swine manure, water solubility was determined by the method of Sharpley and Moyer (2000), and total P by a modified semimicro-Kjeldahl procedure (Bremner, 1996). Following dry matter determination of manures, approximately 1 g (dry weight) of field moist manure was shaken for 1 h in 200 mL distilled water. The supernatant was then filtered (Whatman No. 1 paper) and P determined in the filtrate by the colorimetric method of Murphy and Riley (1962).

Runoff Plots and Rainfall Simulation

At the three sites, 10 surface runoff plots, 2 m long and 1 m wide (long axis orientated down slope), delineated by metal borders installed 5 cm above and below ground level were installed (two replicates of five treatments for a total of 10 plots). The four P sources were broadcast at rate 100 kg P ha^{-1} (based on the content of total P) to duplicate runoff plots; the fifth pair of plots were controls with no P added.

Rainfall was applied 1 d before and 1 d after application of P sources, with one TeeJet 2HH-SS30WSQ nozzle approximately 2.5 m above the soil to achieve terminal velocity (Humphry et al., 2002; National Phosphorus Research Project, 2006). The nozzle, associated plumbing, in-line filter, pressure gauge, and electrical wiring were mounted on a 3- by 3- by 3-m aluminum frame, fitted with canvas walls to provide a windscreen (Fig. 2). A coefficient of uniformity $>85\%$ was obtained for rainfall over a 4-m^2 footprint, which encompasses one pair of abutting plots. An average rainfall intensity of 50 mm h^{-1} was applied

until 30 min of runoff was obtained. This rainfall intensity and duration has an approximate 5-yr return frequency in south-central Pennsylvania. Local ground water was used as the water source for the simulator, and had a DRP concentration of $<0.01 \text{ mg L}^{-1}$, TP of 0.02 mg L^{-1} , nitrate N of 3.1 mg L^{-1} , pH of 5.7, and electrical conductivity of 0.02 S m^{-1} .

Before rainfall, volumetric soil moisture (θ) was determined using a capacitance sensor at five locations within a plot (Theta Probe, Delta-T Devices, Cambridge, UK). Five surface soil samples (0 to 5 cm depth) were collected adjacent to each plot before rainfall and P source application. Soils were air-dried, sieved (2 mm), and stored for chemical analysis.

Surface runoff was collected in metal gutters at the down slope edge of each plot and pumped to 200-L plastic containers. Total surface runoff was measured by weighing the containers. A runoff sample was collected from each container after thorough mixing and agitation, and a subsample was immediately filtered ($0.45 \mu\text{m}$) and stored at 277 K. Filtered samples were analyzed within 24 h of collection and unfiltered samples no more than 7 d after the completion of the rainfall simulation.

All methods that were used in plot design and installation, rainfall simulation and runoff collection, and analysis, followed protocols detailed in the National Phosphorus Research Project (2006). Between May and June 2005 rainfall was applied to each plot 7, 21, and 42 d later and runoff collected. The plots were covered with plastic sheeting when rain was forecast, to eliminate natural rainfall and runoff.

Soil and Water Analysis

Soil pH was determined with a glass electrode at a 1:1 soil/distilled water ratio. Soil cation exchange capacity (CEC) was determined with 1 M NH_4OAc adjusted to pH 7.0 (Hendershot et al., 1993). Extractable Fe and Al were determined by shaking 0.5 g of soil in 20 mL of 1 M NH_4OAc (adjusted to pH 4.8) for 2 h (Olson, 1965; McLean, 1965). Particle size analysis was conducted by the hydrometer method after dispersion with sodium hexametaphosphate (Day, 1965). Mehlich-3 extractable soil P was determined on each soil sample by shaking 1 g of soil with 10 mL of 0.2 M CH_3COOH , 0.25 M NH_4NO_3 , 0.015 M NH_4F , 0.013 M HNO_3 , and 0.001 M EDTA for 5 min (Mehlich, 1984). Water-extractable soil P was measured by shaking 1 g of soil in 10 mL of distilled water for 1 h. Total P was analyzed by modified semimicro Kjeldahl procedure (Bremner, 1996). In all cases, P concentrations were determined on filtered (Whatman No. 1 paper) and neutralized extracts and digests by the colorimetric method of Murphy and Riley (1962).

Soil P sorption was determined by shaking 1 g of soil with various additions of P (0 to 1000 mg kg^{-1} added as KH_2PO_4) in 25 mL of 0.01 M CaCl_2 on an end-over-end shaker at 298 K (Nair et al., 1984). After 24 h, the soil suspensions were centrifuged and filtered (Whatman No. 1 paper) and the solution P concentration (C) determined. The amount of P sorbed (X) is the difference between P added and P remaining in solution. Using the Langmuir sorption equation soil P sorption and soil P sorption maximum were calculated as the reciprocal of the slope of the plot C/X vs. C (Olsen and Watanabe, 1957; Syers et al., 1973).

Dissolved reactive P was determined on filtered ($0.45 \mu\text{m}$) runoff by the colorimetric method of Murphy and Riley (1962). Total P was determined on an unfiltered sample following acid persulfate digestion (Kuo, 1996). All analyses were conducted in duplicate. The loss of DRP and TP from each plot was calculated by multiplying the plot's total runoff volume and respective DRP and TP concentration.

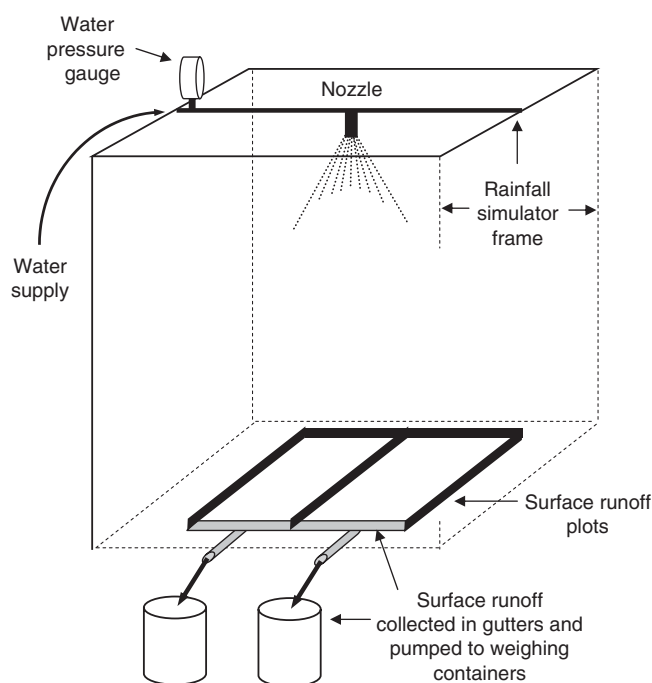


Fig. 2. Schematic representation of the portable rainfall simulator, showing paired 1 by 2 m surface runoff plots, and water collection system. The nozzle is situated approximately 245 cm above the plots.

Table 1. Properties of Alvira, Berks, and Watson soils used in this study.

Soil	Mehlich-3 P	Water ext. P	Total P	CEC†	pH	C	Al	Fe	P sorption max	Binding energy	Sand	Silt	Clay
	mg P kg ⁻¹			meq 100 g ⁻¹		g kg ⁻¹			mg P kg ⁻¹	L mg P ⁻¹	%		
Alvira	44	4.2	646	12.1	6.5	20	17.8	97.7	286	0.36	35	36	29
Berks	69	3.0	457	10.7	6.3	17	23.7	22.5	242	0.17	48	34	18
Watson	72	4.1	664	13.1	6.6	22	19.6	71.2	372	0.35	31	41	28

† CEC, cation exchange capacity.

Statistical Analysis

Statistical analyses were performed using the Statistical Analysis System (SAS Institute, 2001). Data were log (x+1) transformed, as necessary to satisfy conditions of normality using Proc Univariate (SAS Institute, 2001). Analysis of variance (ANOVA) techniques were used to determine treatment effects. Comparison of means were performed using Duncan's studentized range test at $P < 0.05$ (SAS Institute, 2001).

RESULTS AND DISCUSSION

Soil and Phosphorus Sources Properties

The physical and chemical soil properties of the three selected soils are presented in Table 1. Few differences between soils were observed, although Berks had a slightly lower clay content, lower P sorption max, and binding energy than Alvira and Watson soils. Both total and water-extractable soil P concentrations were lower in the Berks than Alvira and Watson. There were large differences in P source WSP, which increased in the order NCRP, swine manure (SM), LGSSP, and TSP (Table 2). However, NCRP had a higher content of TP (133 g kg⁻¹) than LGSSP (79 g kg⁻¹) and SM (1 g kg⁻¹) (Table 2).

Phosphorus Concentrations in Surface Runoff

The mean concentrations of DRP and TP in surface runoff obtained from the rainfall simulation 1 d before source P application were similar among plots of the same soil type at the three sites used in this study. Concentrations of DRP from Alvira, Berks, and Watson soils averaged 0.40, 0.46, and 0.38 mg L⁻¹, respectively, and 0.43, 0.62, and 0.40 mg L⁻¹ for TP. The similarity of these concentrations is consistent with initial surface soil Mehlich-3 P concentrations, which were similar among the 3 soil sites (Table 1). Because P sources were broadcast on vegetated plots, source WSP masked any influence of differing P level (Mehlich-3 P of 44 to 72 mg kg⁻¹) and sorption properties (P sorption maximum of 242 to 372 mg kg⁻¹) of surface soil on runoff P. As a result, soil type had no effect on P loss under the field conditions used in the research and P source was the main variable influencing observed DRP and TP concentrations.

The surface application of all P sources increased the concentration of DRP and TP in runoff 1 d after fertilizer application (Table 3). The general increase in runoff DRP and TP concentrations with application of P sources, considering all soils vs. sources, was consistent and significant ($P < 0.05$) with the increase in source P solubility. For instance, DRP averaged 0.27, 0.50, 14.66, 41.69, and 90.47 mg L⁻¹, and total P 0.34, 0.61, 19.05, 43.1, and 98.06 mg L⁻¹ for the control, NCRP, SM, LGSSP, and TSP, respectively, with most DRP ranging from 76 to 96% for amended plots and 79% for the control plot.

In this study, 79% of the TP from TSP used was water-soluble, contrasting with 50, 30, and 0.5% for LGSSP, SM, and NCRP respectively (Table 2). Although all P sources were applied at the same rate of 100 kg TP ha⁻¹, LGSSP, SM, and NCRP resulted in lower soluble P applications. Several factors can affect the potential for P loss from soils amended with P. For instance, the water solubility of P in both soil and added sources has a large influence on DRP concentrations in runoff (McDowell and Sharpley, 2001). In fact, DeLaune et al. (2004) observed that DRP concentrations in runoff were proportional to the water-soluble P content of applied P sources. Tabbara (2003) found comparatively lower P concentrations in runoff from a fallow sandy loam soil after application of liquid swine manure than liquid ammonium polyphosphate fertilizer and attributed this to the lower concentration of soluble P in applied manure than fertilizer.

Although all P sources increased DRP and TP concentrations of runoff ($P < 0.05$), NCRP application resulted in only small, albeit significant increases in DRP and TP (Table 3). This slight increase in runoff P following NCRP amendment compared with the control was probably due the low WSP and consequently slow release of P from NCRP. This reflects the slow RP dissolution kinetics (Di et al., 1994) and that the potential for P release from NCRP in a short period after application (1 d) being much lower than from the other P sources. McDowell et al. (2003), evaluating the effect of applying 23 kg P ha⁻¹ yr⁻¹ as reactive rock phosphate (RRP) or 20 kg P ha⁻¹ yr⁻¹ as superphosphate on P transport in runoff caused by a 15 mm h⁻¹ rainfall for

Table 2. Properties of the P sources used in this study.

P source	Total P	Water-soluble P	Citrate-EDTA + H ₂ O soluble P	Moisture content
	g kg ⁻¹	% total P	g kg ⁻¹	%
North Carolina rock P	133	0.6	16.4	12
Low-grade single super P	79	39.9	54.9	70
Triple super P	201	159.0	185.3	92
Swine manure	1.0	0.3	ND†	ND

† ND, data not determined.

Table 3. Dissolved reactive phosphorus (DRP) and total P (TP) concentration in surface runoff from the three soils used in this study.[†]

Source	1 d after P application		7 d after P application		21 d after P application		42 d after P application	
	DRP	TP	DRP	TP	DRP	TP	DRP	TP
	mg L^{-1}							
Control	0.27e	0.34d	0.37d	0.46c	0.38c	0.48c	0.37c	0.44c
NCRP [‡]	0.50d	0.61d	0.50c	0.62c	0.64b	0.74b	0.53b	0.97b
LGSSP	41.69b	43.10b	3.23b	3.92b	2.01a	2.36a	1.24a	1.92a
TSP	90.47a	98.06a	7.73a	8.12a	2.94a	3.38a	1.96a	2.06a
SM	14.66c	19.05c	5.11ab	5.47ab	3.25a	3.34a	1.97a	2.40a

[†] Values within a column followed by the different letters are significantly different ($P < 0.05$) as determined by comparison of means.

[‡] NCRP, North Carolina rock phosphate; LGSSP, "low-grade" single superphosphate; TSP, triple superphosphate; SM, swine manure.

1 h, measured average DRP concentrations of 0.11 and 0.49 mg L^{-1} after respective RRP and superphosphate applications and TP concentrations of 0.30 and 1.00 mg L^{-1} . As in the present study, Nguyen et al. (1999) found DRP concentrations in runoff were greater from soil amended with high water-soluble P sources, SSP, and TSP than from soils amended with slow-release RRs from Tunisia (Gafsa) and Egypt (Kosseir) that were only slightly greater than the control 3 d after the application (Nguyen et al., 1999).

Twenty-one days after P source application, DRP and TP in surface runoff did not differ significantly among TSP, LGSSP, and SM ($P < 0.05$). However, these sources showed higher DRP and TP in surface runoff compared with NCRP and the control (Table 3). This trend was also observed 42 d after fertilizer application (Table 3). When compared with the first day after P source application, concentrations were low, with a dramatic decrease in DRP and TP concentrations for all sources except NCRP. The constant concentration of DRP and TP in runoff with time after NCRP application is related to the slow dissolution kinetics of the RP.

Phosphorus Loss in Surface Runoff

Phosphorus source did not affect runoff volume over the study period (Table 4). However, average cumulative DRP and TP losses ranged from 0.45 to 20.93 and 0.74 to 22.71 kg ha^{-1} , respectively (Table 4). Analysis of

variance showed that P source had a significant effect (P of 0.05) on cumulative DRP and TP losses, with greater losses from plots fertilized with TSP (79% WSP). Cumulative TP losses increased as a function of source WSP (TSP>LGSSP>SM>NCRP; Table 4). A simple regression analysis showed that both DRP and TP concentrations in runoff were a strong positive linear function ($r^2 = 0.99$) of P source WSP (Fig. 3). These results are consistent with those of Franklin et al. (2006) who found larger cumulative losses of DRP for plots fertilized with TSP (95% WSP) than plots fertilized with broiler litter (8% WSP).

Total P loss in runoff accounted for 23, 13, 7, and 0.7% of the TP applied as TSP, LGSSP, SM, and NCRP, respectively (Table 4). These results show that differences in cumulative loss in DRP and TP were strongly impacted by source type, as a function of its WSP concentration. Further, DRP constituted the main proportion of TP loss, accounting for 92% in runoff from plots amended with TSP, 94% with LGSSP, 83% with SM, and 61% with NCRP.

CONCLUSIONS

Results from this research clearly show the strong influence of water solubility of P sources in determining P concentration and loss in surface runoff. A drastic increase in DRP, TP, and cumulative loss was observed

Table 4. Surface runoff volume and cumulative losses of dissolved reactive and total P.

Soils	P sources [†]				
	Control	NCRP	LGSSP	TSP	SM
Surface runoff volume, L					
Alvira	158	208	226	185	205
Berks	101	105	205	184	211
Watson	199	159	167	174	173
Average [‡]	153a	158a	200a	181a	196a
Cumulative dissolved reactive P loss, kg ha^{-1}					
Alvira	0.28	0.52	18.08	32.16	5.62
Berks	0.18	0.39	10.78	14.46	5.60
Watson	0.23	0.43	9.18	16.18	6.53
Average [‡]	0.23d	0.45c	12.68a	20.93a	5.92b
Cumulative total P loss, kg ha^{-1}					
Alvira	0.35	0.83	18.36	33.16	6.81
Berks	0.30	0.68	11.92	15.54	6.83
Watson	0.31	0.72	9.99	19.57	7.64
Average [‡]	0.32e	0.74d	13.42b	22.71a	7.09c

[†] TSP, triple superphosphate; LGSSP, "low-grade" single superphosphate; SM, swine manure; NCRP, North Carolina rock phosphate.

[‡] Average runoff volume, DRP, and TP loads followed by the different letters are significantly different ($P < 0.05$) as determined by analysis of variance.

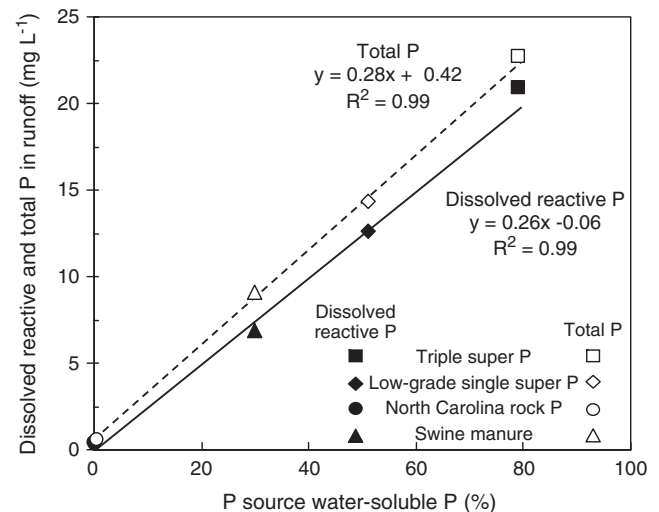


Fig. 3. Relationship between percent of P source total P as water-soluble P and the dissolved reactive and total P concentration of surface runoff averaged for all three soils and rainfall-runoff events.

with an increased source of WSP (TSP>LGSSP>SM>NCRP>control). Phosphorus concentration and loss in runoff from NCRP was only slightly greater than that from the control, as NCRP contributed a slow but continued release of P over the time of this field study. Phosphorus concentrations in runoff were highest in the first rainfall event 1 d after P source application and decreased with time. These results support the use of P source WSP as an indicator of the potential for these sources to enrich runoff P after being applied to agricultural land.

These results also support the use of P solubility for mineral fertilizers as well as manures in the P Index, in terms of assessing the potential risk of P loss from agricultural lands. In addition, the results of this research suggest that decisions regarding the source of P or type of fertilizer applied can influence environmental as well as agronomic response. For instance, whereas TSP provides a ready and large amount of P immediately available for plant uptake, it is also an immediate source of P to runoff. With time after TSP application, P enrichment of runoff rapidly declines due to depletion of this P source. In contrast, following application of a lower P solubility source of P, such as LGSSP or NCRP, runoff P enrichment is not as great as with TSP. However, the low WSP of LGSSP and NCRP can maintain a slow release of P, such that 42 d after application, P loss from LGSSP- and NCRP-amended plots was similar to that from TSP.

Clearly, a decision can be made regarding P source use based on expected environmental or agronomic response. Whereas other studies have reported on the relative agronomic effectiveness of different types of P fertilizers, our research suggested that more agronomically effective P sources present a greater risk of runoff P enrichment. Conversely, LGSSP and NCRP may support lower concentrations of P in runoff than TSP. Nevertheless, if crop is the main management consideration, then agronomic efficiencies should determine the source or type of P applied.

ACKNOWLEDGMENTS

The authors would like to thank the staff of the USDA-ARS, Pasture Systems and Watershed Management Research Unit, University Park and Klingerstown, PA, for their valuable contribution to this study: Todd Strohecker, Mike Reiner, and Jim Richards who helped conduct rainfall simulations; Mary-Kay Lupton, Joan Weaver, and Bart Moyer for the help and suggestions during soil and water analyses; Lou Saporito for the support on the statistical analysis; and CAPES and CNPq for funding Francirose Shigaki to conduct this research.

REFERENCES

Association of Official Analytical Chemists. 1995. Fertilizers–Phosphorus. p. 1–12. In P. Cunniff (ed.) Official Methods of analysis of AOAC International, 16th edition, V.11: Agricultural Chemicals; Contaminants; Drugs. AOAC International, Arlington, VA.

Bremner, J.M. 1996. Nitrogen–Total. In D.L. Sparks (ed.) Methods of soil analysis. Part 3. Soil Sci. Soc. Am. Book Series 5:1085–1121. SSSA, Madison, WI.

Burkholder, J.A., and H.B. Glasgow, Jr. 1997. *Pfiesteria piscicidia* and

other *Pfiesteria*-dinoflagellates behaviors, impacts, and environmental controls. *Limnol. Oceanogr.* 42:1052–1075.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecol. Appl.* 8:559–568.

Chien, S.H., and R.G. Menon. 1995. Factors affecting the agronomic effectiveness of phosphate rock for direct application. *Fert. Res.* 41: 227–234.

Day, P.R. 1965. Particle fractionation and particle size analysis. In C.A. Black et al. (ed.) Methods of soil analysis. Part 1. Agronomy Monograph 9:545–567. ASA, Madison, WI.

DeLaune, P.B., P.A. Moore, Jr., D.K. Carman, A.N. Sharpley, B.E. Haggard, and T.C. Daniel. 2004. Evaluation of the phosphorus source component in the phosphorus index for pastures. *J. Environ. Qual.* 33:2192–2200.

Di, H.J., R. Harrison, and A.S. Campbell. 1994. Assessment of methods for studying the dissolution of phosphate fertilizers of differing solubilities in soil. I. An isotopic method. *Fert. Res.* 38:1–9.

Franklin, D.H., M.L. Cabrera, and V.H. Calvert. 2006. Fertilizer source and soil aeration effects on runoff volume and quality. *Soil Sci. Soc. Am. J.* 70:84–89.

Hendershot, W.H., H. Lalonde, and M. Duquette. 1993. Ion exchange and exchangeable cations. p. 167–176. In M.R. Carter (ed.) Soil sampling and methods of analysis. Lewis Publishers, Boca Raton, FL.

Humphry, J.B., T.C. Daniel, D.R. Edwards, and A.N. Sharpley. 2002. A portable rainfall simulator for plot-scale runoff studies. *Appl. Eng. Agric.* 18:199–204.

Kleinman, P.J.A., A.N. Sharpley, B.G. Moyer, and G.F. Elwinger. 2002. Effect of mineral and manure phosphorus sources on runoff phosphorus. *J. Environ. Qual.* 31:2026–2033.

Kuo, S. 1996. Phosphorus. In D.L. Sparks (ed.) Methods of soil analysis, Part 3. Chemical methods. Soil Sci. Soc. Am. Book Series 5:869–919. SSSA, Madison, WI.

Leon, L.A., W.E. Fenster, and L.L. Hammond. 1986. Agronomic potential of eleven phosphate rocks from Brazil, Colombia, Perú, and Venezuela. *Soil Sci. Soc. Am. J.* 50:798–802.

McDowell, R.W., R.M. Monaghan, and P.L. Carey. 2003. Potential phosphorus losses in overland flow from pastoral soils receiving long-term applications of either superphosphate or reactive phosphate rock. *N. Z. J. Agric. Res.* 46:329–337.

McDowell, R.W., and A.N. Sharpley. 2001. Approximating phosphorus release from soils to surface runoff and subsurface drainage. *J. Environ. Qual.* 30:508–520.

McLean, E.O. 1965. Aluminum. p. 963–973. In C.A. Black et al. (ed.) Methods of soil analysis Part 2. ASA, Madison, WI.

Mehlich, A. 1984. Mehlich-3 soil test extractant: A modification of Mehlich-2 extractant. *Commun. Soil Sci. Plant Anal.* 15:1409–1416.

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural water. *Anal. Chim. Acta* 27:31–36.

Nair, P.S., T.J. Logan, A.N. Sharpley, L.E. Sommers, M. Tabatabai, and T.L. Yuan. 1984. Interlaboratory comparison of a standardized phosphorus adsorption procedure. *J. Environ. Qual.* 13:591–595.

National Phosphorus Research Project. 2006. National phosphorus research project for simulated rainfall-surface runoff studies. Southern Extension-Research Activity- Information Exchange SERA-IA. Available at http://www.sera17.ext.vt.edu/Documents/National_P_protocol.pdf (accessed 13 Mar. 2006; verified 4 Aug. 2006).

National Research Council. 2000. Clean coastal waters: Understanding and reducing the effects of nutrient pollution. National Academy Press, Washington, DC.

Nguyen, M.L., A.B. Cooper, and B.S. Thorrold. 1999. Phosphorus losses in surface runoff from field plots receiving soluble and slow-release phosphatic fertilizers. NIWA Client Rep. SQF90201. National Institute of Water and Atmospheric Research, Hamilton, New Zealand.

Olsen, S.R., and F.S. Watanabe. 1957. A method to determine a phosphorus adsorption maximum of soils as measured by the Langmuir isotherm. *Soil Sci. Soc. Am. Proc.* 21:144–149.

Olson, R.V. 1965. Iron. p. 978–998. In C.A. Black et al. (ed.) Methods of soil analysis, Part 2. Agron. Monogr. 9. ASA, Madison, WI.

Prochnow, L.I., S.H. Chien, G. Carmona, and J. Henao. 2004. Greenhouse evaluation of phosphorus sources produced from a low-reactive Brazilian phosphate rock. *Agron. J.* 96:761–768.

- Prochnow, L.I., S.H. Chien, R.W. Taylor, G. Carmon, J. Henao, and E.F. Dillard. 2003. Characterization and agronomic evaluation of single superphosphates varying in iron phosphate impurities. *Agron. J.* 95:293–302.
- SAS Institute. 2001. SAS OnlineDoc. Version 8.02. SAS Inst., Cary, NC.
- Sharpley, A.N. 2003. Plowing to decrease surface stratification of phosphorus in manured soils. *J. Environ. Qual.* 32:1375–1384.
- Sharpley, A.N., and B. Moyer. 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. *J. Environ. Qual.* 29:1462–1469.
- Sharpley, A.N., J.K. Syers, and P.E.H. Gregg. 1978. Transport in surface runoff of phosphorus derived from dicalcium phosphate and superphosphate. *N. Z. J. Sci.* 21:307–310.
- Sharpley, A.N., J.L. Weld, D.B. Beegle, P.J.A. Kleinman, W.J. Gburek, P.A. Moore, and G. Mullins. 2003. Development of phosphorus indices for nutrient management planning strategies in the U. S. *J. Soil Water Conserv.* 58:137–152.
- Sims, J.T., B.C. Joern, and R.R. Simard. 1998. Phosphorus losses in agricultural drainage: Historical perspective and current research. *J. Environ. Qual.* 27:277–293.
- Syers, J.K., M.G. Browman, G.W. Smillie, and R.B. Corey. 1973. Phosphate sorption by soils evaluated by the Langmuir adsorption equation. *Soil Sci. Soc. Am. Proc.* 37:358–363.
- Tabbara, H. 2003. Phosphorus loss to runoff water twenty-four hours after application of liquid swine manure or fertilizer. *J. Environ. Qual.* 32:1044–1052.
- USDA, and USEPA. 1999. Unified national strategy for animal feeding operations. U.S. Gov. Print. Office, Washington, DC. Available at http://cfpub.epa.gov/npdes/afo/ustrategy.cfm?program_id=7 (accessed 1 Sept. 2005; verified 4 Aug. 2006).
- USEPA. 1996. Environmental indicators of water quality in the United States. EPA 841-R-96-002, U.S. Gov. Print. Office: Washington, DC.
- U.S. Geological Survey. 1999. The quality of our nation's waters: Nutrients and pesticides. U.S. Geological Survey Circular 1225. USGS Information Services, Denver, CO. Available at <http://www.usgs.gov> (accessed 13 Mar. 2006; verified 4 Aug. 2006).
- Withers, P.J.A., S.D. Clay, and V.G. Breeze. 2001. Phosphorus transfer in runoff following application of fertilizer manure and sewage sludge. *J. Environ. Qual.* 30:180–188.